

Fire in the Southeastern Deserts Bioregion

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Because of the inescapably close correlation between prevalence of fire and amount of fuel, deserts are characteristically less affected by fire than are most ecosystems ... however, even though fire frequency and severity may be relatively low in any rating scale, their effect on the ecosystem may be extreme.
- Humphrey 1974, page 366.

Description of Bioregion

Physical Geography

The southeastern deserts bioregion (desert bioregion) occupies the southeastern 27% of California (110,283 km² or 27,251,610 acres) (Miles and Goudy 1997). The desert bioregion is within the basin and range geomorphic province of western North America, and includes two ecoregional provinces comprised of five ecological sections. The American Semi-Desert and Desert Province (hot-desert province) includes the Mojave Desert, Sonoran Desert, and Colorado Desert sections in the southern 83% of the desert bioregion (Table 16.1). The Intermountain Semi-Desert Province (cold desert province) includes the Southeastern Great Basin and Mono sections in the northern 17% of the desert bioregion.

Table 16.1. General descriptions and lightning frequencies (1985-2001) in the ecological sections of the southeastern deserts bioregion.

Ecological section ^a	% of bioregion	Constituent ecological zones ^b	Predominant Küchler vegetation types ^c	Lightning strikes/100km ² /yr ^d
Mojave	61	low, mid, high, montane, riparian	desert shrub 58% barren 37%	30
Sonoran	12	low, riparian	barren 82% desert shrub 18%	25
Colorado	10	low, mid, riparian	desert shrub 57% barren 38%	12
SE Great Basin	10	high montane, riparian	desert shrub 74% juniper-pinyon 18%	29
Mono	7	high, montane, riparian	sagebrush 46% juniper-Pinyon 15%	32

^a Miles and Goudy (1997)

^b low elevation desert shrubland, middle elevation desert shrubland and grassland, high elevation desert shrubland and woodland, desert montane woodland and forest, desert riparian woodland and oasis (see detailed descriptions in the text)

^c potential natural vegetation types (Küchler 1964) that constitute 15% or more of the ecological section.

^d Bureau of Land Management lightning detection data (van Wagendonk and Cayan, in press)

The geomorphology of the desert bioregion is characterized by isolated mountain ranges with steep slopes separated by broad basins containing alluvial fans, lava flows, dunes, and playas. Elevations range from -85 m (-280 ft) below sea level in Death Valley, to 4,328 m (14,200 ft) above sea level in the White Mountains. Soil taxa range widely from hyperthermic or thermic, aridic Aridisols and Entisols in the Colorado, Sonoran, and Mojave Desert sections, to thermic, mesic, frigid, or cryic, aridic, xeric, or aquic Alfisols, Aridisols, Entisols, Inceptisols, Mollisols, and Vertisols in the Mono and Southeastern Great Basin sections (Miles and Goudy 1997). This wide range in geomorphology and soil conditions translates into a wide range of vegetation and fuel types, which include arid shrublands and semi-arid shrublands, grasslands, woodlands, and forests.

Climatic Patterns

Although frontal cyclones of the jet stream pass through the region during winter (November through April), virtually the entire desert bioregion is arid due to rain shadows of the Sierra Nevada, Transverse, and Peninsular ranges (Chapter 2, this volume). Precipitation locally increases with orographic lift in desert ranges, particularly those that rise above 2,000 m (6,096 ft). From July to early September, the region experiences 10 to 25 days of afternoon thunderstorms from the North American monsoon originating in the Gulf of California and Mexico. Thunderstorm cells tend to concentrate over high terrain, especially the eastern escarpments of the Sierra Nevada, Transverse, and Peninsular ranges, in the mountains of the eastern Mojave Desert, and in the high basin and range terrain between the White Mountains and Death Valley. The average annual precipitation on valley floors ranges from 10 to 20 cm (3.9 to 7.9 in) in the Mojave Desert and southeastern Great Basin, to 7 to 10 cm (2.8 to 3.9 in) in the Colorado and Sonoran deserts. The average annual rainfall total at Death Valley (5.8 cm, 2.3 in) is the lowest in North America. Precipitation increases to 20 to 30 cm (7.9 to 11.8 in) in the mountains above 2,000 m (6,562 ft), 40 cm (15.8 in) in the White Mountains, and 60 cm (23.6 in) in the upper leeward catchments of the Sierra Nevada, Transverse, and Peninsular ranges. The percentage of annual precipitation falling during summer (May through October) ranges from approximately 20% in the southeastern Great Basin to 40% at the Colorado River in the Sonoran Desert.

Interannual variation in rainfall is relatively high compared to other California bioregions, resulting in highly variable frequency and extent of fires among years. High rainfall produces fine fuels that promote fire spread, especially in the hot desert sections where fuels are otherwise sparse. Low rainfall causes shrub mortality which reduces woody fuel moisture and may promote fire spread in the cold desert sections where woody fuel cover is relatively high, although low fine fuel loads caused by low rainfall is probably more limiting to fire spread. Multi-decadal variation in rainfall has also been significant, with periods of relatively high rainfall from the turn of the century until 1946, a mid-century drought from 1947 to 1976, and a period of high rainfall 1977 to 1998 (Hereford et al. in press). This approximately 30-year cycle, coupled with below-average rainfall from

1999 to 2004, suggest that another 30-year drought period may be establishing, which could lead to reduced frequency and size of fires in most of the desert bioregion entering the 21st Century.

The entire desert bioregion has a large annual range of temperature due to its isolation from the stabilizing influences of the Pacific Ocean. There is also large regional variability due to variable elevational relief. Average January temperatures on valley floors range from -3 to 0° C (27 to 32° F) in the northeastern Great Basin to 7 to 10° C (45 to 50° F) in the Mojave Desert, and 11 to 13° C (52 to 55° F) in the Sonoran and Colorado deserts. Temperatures decrease with altitude to about 0° C (32° F) at 2,000 m (6,562 ft) and -8° C (18° F) at 3,000 m (9,842 ft.). During summer, average temperatures vary near the dry adiabatic lapse rate due to intense atmospheric heating in the absence of evapotranspiration under high rates of insolation. July average temperatures on valley floors range from 18 to 20° C (64 to 68° F) in the northeastern Great Basin to 25 to 30° C (77 to 86° F) in the Mojave Desert and 30 to 35° C (86 to 95° F) in the Sonoran and Colorado deserts. Maximum temperatures average > 40° C (104° F) below 1,000 m (3,281 ft) elevation and occasionally reach 50° C (122° F) in Death Valley, the Colorado River, and the Salton Sea trough. In the desert mountains, average temperatures decrease to 20° C (68° F) at 2,000 m (6,562 ft) and 10° C (50° F) at 3,000 m (9,842 ft). The decrease in temperature with altitude results in rapid decrease in evapotranspiration which in phase with increasing precipitation results in corresponding increase in woody biomass of ecosystems. Light snowpacks 10 to 15 cm (3.9 to 5.9 in) deep can develop in winter but typically disappear by spring above 2,000 m (6,562 ft), although deeper snow of 100 cm (39.4 in) can persist into the spring in subalpine forests > 3,000 m (9,842 ft).

Relative humidity during the afternoon in the summer fire season, when fires are most likely to spread, is very low throughout the desert bioregion. Average relative humidity in July ranges from 20 to 30% in the northeastern Mojave Deserts to 10 to 20% in the Mojave, Sonoran, and Colorado deserts. Values are low because moisture of the Pacific Coast marine layer is mixed aloft with dry subsiding air masses upon dissipation of the marine inversion, as well as from high temperatures produced by convective heating of surface air layers. The lowest humidity of the year (frequently < 10%) typically occurs in late June, just before the arrival of the North American monsoon.

Lightning frequency is higher in the desert than in any other California bioregion (van Wagendonk and Cayan in press). Lightning strikes/100km²/year averaged 27 (sd = 16) from 1985 through 2000, ranging from 32 in the Mono to 12 in the Colorado Desert sections (Table 16.1). The bioregions with the next most frequent lightning strikes were the Northeast Plateau (22 strikes/100km²/year) and Sierra Nevada (20 strikes/100km²/year). Most lightning in the desert bioregion occurred from July through September (78%), resulting from summer monsoons which developed in the Colorado, Sonoran, and eastern Mojave deserts, and from summer storms that developed in the Sierra Nevada mountains and drifted into the southeastern Great Basin and Mono sections. Lightning also occurred primarily during daylight hours, with 81% between 0600 and 1800.

Ecological Zones

From a fire ecology perspective, much of the variation in the desert bioregion relates to patterns of fuel characteristics and fire regimes. Vegetation (fuels), topography, and lightning strikes per unit area vary locally with elevation, and elevational vegetation gradients are correlated positively with latitudinal gradients and ecotones with more mesic regions in the immediate rain shadow of the Sierra Nevada, Transverse, and Peninsular ranges. Accordingly, we consider elevation to be the primary determinant of fire ecology zones in the desert bioregion. The ecological zones described below are listed in order of increasing elevation, except for the riparian zone, which transcends many of the other zones.

Low elevation desert shrubland zone

This is the predominant ecological zone in the Sonoran Desert section. Major vegetation types include alkali sink vegetation and the lower elevations of creosote bush scrub (Munz and Keck 1959) and succulent scrub (Rowlands 1980). Surface fuel loads and continuity are typically low, hindering the spread of fire (Fig 16.1).



Fig.16. 1. The low elevation desert shrubland ecological zone. This photo shows a creosotebush scrub vegetation typical of the Sonoran Desert.

Middle elevation desert shrubland and grassland zone

This is the predominant ecological zone in the Mojave Desert, Colorado Desert, and Southeastern Great Basin sections, where it typically occurs as an elevational band above the low elevation zone and below the high elevation zone. It also occurs at the regional ecotone between the Mojave and Great Basin deserts. Major vegetation types include Joshua tree woodland, shadscale scrub, the upper elevations of creosote bush scrub (Munz and Keck 1959), blackbrush scrub, and desert scrub-steppe (Rowlands 1980). Surface fuel characteristics are variable, but loads and continuity can be relatively high compared to the low elevation zone, facilitating the spread of fire (Fig. 16.2).



Fig. 16.2. The middle elevation desert shrubland and grassland ecological zone. This photo shows a blackbrush scrubland, which typically includes blackbrush, Mojave yucca, Joshua tree, and California juniper.

High elevation desert shrubland and woodland zone

This is the predominant ecological zone in the Mono section. It also occurs at the tops of most Mojave Desert mountains or just below desert montane forests, and along the margins of the Sierra Nevada, Transverse, and Peninsular mountain ranges where they intergrade with yellow pine forests. Major vegetation types include sagebrush scrub, pinyon-juniper woodland, and desert chaparral (Munz and Keck 1959). Surface fuel loads and continuity are high where sagebrush scrub and chaparral dominate, facilitating the spread of fire. However, surface fuels are replaced by very high loads of crown fuels in closed pinyon-juniper woodlands, where fires only occur under extreme fire weather conditions and are typically very intense (Fig. 16.3).



Fig. 16.3. The high elevation desert shrubland and woodland ecological zone. This photo shows a pinyon-juniper woodland.

Desert montane woodland and forest zone

This zone is very limited in total area, and occurs almost exclusively in the Mono and Southeast Great Basin sections. Major vegetation types include bristlecone pine forest and alpine fell-fields (Munz and Keck 1959). Surface fuels are typically sparse, separating patches of crown fuels and hindering the spread of fire (Fig. 16.4).



Fig 16.4. The desert montane woodland and forest ecological zone. This photo shows a bristlecone pine forest.

Desert riparian woodland and oasis zone.

This zone includes a diverse set of vegetation types that do not fit into any single elevational range. Vegetation types include oases and riparian woodlands, shrublands, grasslands, and marshes. Surface fuels loads and continuity can be very high, facilitating fire spread, although vertical continuity of ladder fuels and horizontal continuity of crown fuels are often insufficient to carry crown fires (Fig. 16.5).



Fig. 16.5. The desert riparian woodland and oasis ecological zone. This photo shows a riparian shrubland and woodland.

Overview of Historic Fire Occurrence

The primary factor controlling fire occurrence in the desert bioregion is fuel condition, specifically fuel continuity and fuel type. Where fuel continuity is low, as in most of the low elevation and desert montane ecological zones, fires will not typically spread beyond ignition points. Even where continuity is relatively high, fuelbeds may be comprised primarily of fuel types that do not readily burn except under the most extreme fire weather conditions. The coarse, woody fuels of pinyon-juniper woodlands in the high elevation ecological zone are a good example. Thus, variations in fuel condition are central to any attempts to evaluate past or current patterns of fire occurrence.

Prehistoric

Prehistoric fire regimes have not been quantitatively described for most of the desert regions of southwestern North America, largely because the usual tools for reconstructing fire histories, such as analyzing trees for fire scars or coring sediments in swamps or lakes for charcoal deposits, cannot be used where the requisite trees or lakes are not present. As a result, past fire regimes must be inferred indirectly from prehistoric vegetation studies or current observations and data.

Fossil packrat midden data suggest that most of the desert bioregion has been under arid to semi-arid conditions since the beginning of the Holocene (~10,000 years BP), with pinyon and juniper woodlands on upper slopes and at higher elevations, and low scrub and perennial grasslands in valleys and at lower elevations (Van Devender and Spaulding 1979, Koehler et al. 2005). Most interior basins in the desert bioregion did not support permanent lakes except those receiving runoff from the Sierra Nevada, Transverse, or Peninsular ranges. Thus, the major vegetation types that presently occur in the desert bioregion, and the ecological zones described in this chapter, were likely present in the desert bioregion throughout the Holocene, expanding and contracting relative to each other as they shifted up and down elevational gradients with periods of low and high rainfall.

The low elevation ecological zone probably contained low and discontinuous fuels, hindering fire spread and resulting in low intensity, patchy burns and long fire return intervals. Consecutive years of high rainfall would have increased fine fuel loads and continuity, and may have allowed fire to spread periodically in this ecological zone, especially where rainfall was highest along the western margins of the Mojave and Colorado deserts close to the Transverse and Peninsular mountain ranges.

The middle elevation, high elevation, and riparian zones likely had sufficient perennial plant cover to periodically carry fire in the prehistoric past without significant amounts of fine fuels. Because these fires would have been carried by relatively high cover of perennial shrubs and grasses, they were likely moderate intensity, stand replacing fires, as they typically are today.

Fuels in the desert montane zone were probably discontinuous resulting in small, patchy, and very infrequent surface or passive crown fires. Evidence of this is the presence of the long-lived (>3,000 years), but fire sensitive, bristlecone pine trees (*Pinus longaeva*).

It seems highly probable that fuel conditions and fire regimes have remained relatively constant across the desert bioregion during the Holocene, although their spatial distributions likely varied as the ecotones between vegetation formations shifted with alternating periods of low and high rainfall. Current climate conditions have generally persisted since ~1,440 years BP in the Mojave Desert (Koehler et al. 2005), supporting the supposition that relative distributions of fuel conditions and ecological zones have remained relatively constant during at least the latter part of the Holocene. It is also likely that fuel conditions and fire regimes have changed significantly since the late 1880s due to land use activities and invasions by non-native annual grasses. We discuss these changes in more detail below.

Historic

Livestock grazing can reduce perennial plant cover, especially cover of perennial grasses (Brooks et al. in press), which very likely has led to reduced landscape flammability since grazing began in the desert bioregion during the late 1880s. However, at the same time that fuels were reduced due to grazing, ignitions probably increased as fire came into use by livestock operators to convert shrublands into grasslands and increase forage production, especially in the Mono and middle to high elevations of the Southern Great Basin and Mojave sections. For example, rangelands in southern Nevada, southwestern Utah, and northwestern Arizona were extensively burned during the early 1900s to reduce shrub cover and promote the growth of perennial grasses (Brooks et al. 2003). Similar rangeland burns may have also been implemented in the southern and eastern Mojave Desert and the far western Colorado Desert, where summer rainfall occurs in sufficient amounts to support large stands of perennial grasses. However, most of the southern hot desert regions are too dry to support sufficient native fuels to carry fire, so even if ranchers tried to burn, they may have often been unsuccessful.

Analyses of historical aerial photos from 1942, 1953 to 54, 1968, 1971 to 74, 1998, and 1999 at Joshua Tree National Park indicate that there were periodic fires prior to 1942 (Minnich, 2003), during a 30-year period of relatively high rainfall that lasted until 1946 (Hereford et al. in press). However, most fires were <121 ha (300 acres) with the largest encompassing 607 ha (1,500 acres), and all occurred in the middle and high elevation ecological zones (Minnich, 2003). The spatial clustering of burns in some areas suggests that deliberate burning by humans was practiced, possibly to improve range production for livestock. During the mid-century drought, only three small fires occurred, all during the 1960s and in Joshua tree woodlands of the middle elevation ecological zone. Soon after the drought ended in 1977, fires again became more prevalent, but their size and numbers eclipsed what was observed prior to the mid-century drought. The first was a 2,428 ha (6,000 acre) fire in 1978, and the most recent was a 6,070 ha (15,000 acre) complex of fires that burned over a period of 5 days in 1999, both in the middle and high elevation ecological zones. These recent fires at Joshua Tree National Park were fueled largely by old stands of native trees, shrubs, and perennial grasses, but fire spread was additionally facilitated by stands of the non-native annual grasses red brome (*Bromus madritensis* ssp. *rubens*) and cheatgrass (*Bromus tectorum*), especially where fire passed through previously

burned areas where cover of these grasses was especially high (National Park Service, DI-1202 fire reports).

Current

Records from land management agencies provide information on recent fires that can be used to reconstruct current fire regimes across the desert bioregion. We extracted data from fire occurrence records (DI-1202 reports) archived by the United States Department of the Interior and Department of Agriculture between 1980 and 2001 to create basic summaries for each of the five ecological sections in the California desert (Fig. 16.6, Table 16.2). This 21-year database is too short to capture the full range of potential burning conditions, because it was coincident with a period of above-average rainfall from 1977 to 1998 (Hereford et al. in press). However, it represents the best data available to approximate fire regimes since 1980 in the desert bioregion of California.

Table 16.2. Recent fire history (1980-2001)^a in the ecological sections of the southeastern deserts bioregion.

Ecological section ^b	Total fires	Total Area burned	Fire frequency (fires/1000k m ² /yr)	Annual area burned (ha/1000 km ² /yr)	Fire size (ha/fire)	Human: lightning fires	% of lightning strikes that resulted in fires ^c
Mojave	3158	69110	2.1	47	22	3.6	0.6
Sonoran	175	13217	0.6	47	76	7.5	0.2
Colorado	525	21340	2.2	88	41	44.2	1.8
Mono	1630	49292	9.6	292	30	0.5	2.0
SE Great Basin	90	5460	0.4	23	61	1.0	0.1
TOTAL	5578	158419	2.3	66	28	2.0	

^a fire records (DI-1202 reports) of the Department of the Interior and Department of Agriculture, screened for errors as recommended by Brown et al. (2002).

^b Miles and Goudy (1997)

^c lightning frequency (van Wagendonk and Cayan, in press) per lightning fires.

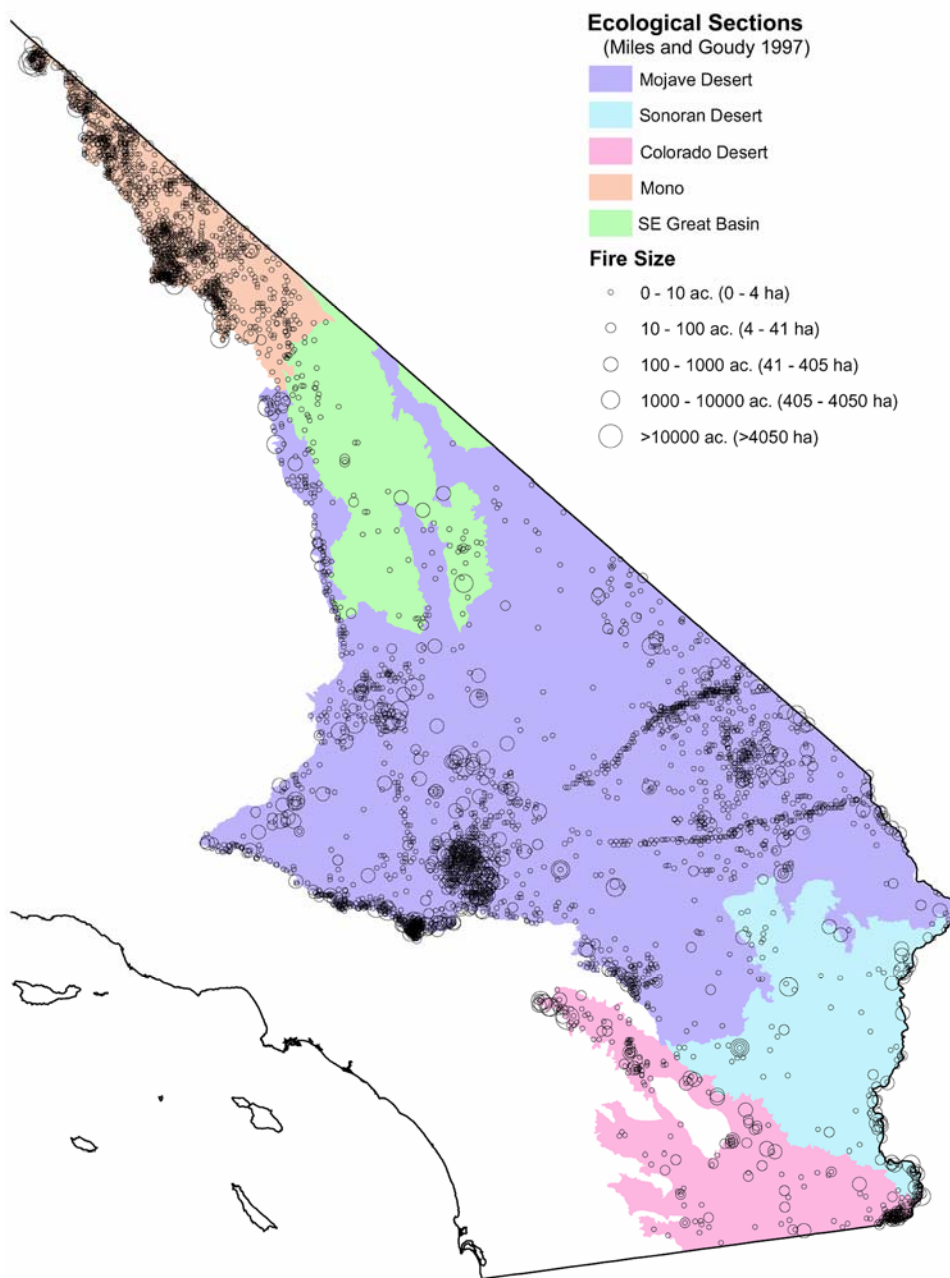


Fig. 16.6. Recent fire occurrences (1980-2001) in the five ecological sections of the Southeastern Deserts Bioregion.

The primary message from these fire records is that the proportion of total area that burned per year from 1980 to 2001 is very small, peaking in the Mono section at 0.3%/yr (292 ha/1,000 km²/yr, Table 16.2), resulting in a fire cycle of 342 years in that ecological section. The annual fire frequency and area burned were highest in the Mono section, and lowest in the southern Great Basin section (Fig 16.6, Table 16.2), peaking from May through September. Among the hot desert regions, fire frequency was highest in the Mojave and Colorado deserts, and the annual area burned was highest in the Colorado Desert. The percentage of lightning strikes that resulted in fire was highest in the Mono and Colorado Desert sections, probably due to high fuel continuity caused by the prevalence of sagebrush steppe in the Mono section, and red brome dominated creosote bush scrub in the western Colorado section. The Colorado Desert section had the highest ratio of human:lightning caused fires. This is probably due to both the high human population density and agricultural activity in the Coachella and Imperial valleys, and the low frequency of lightning in the Colorado Desert (Table 16.1). The northern cold desert regions had the lowest frequency of fires caused by humans, probably due to its remoteness from major human population centers.

In a separate analysis of agency fire data from 1980 to 1995 in the Mojave, Colorado, and Sonoran desert sections, fires were found to be clustered in regional hotspots (Brooks and Esque 2002), where they were much more frequent and burned more proportional area than the desert-wide averages indicated in Table 16.2. Annual fire frequency increased significantly from 1980 through 1995 ($r^2 = 0.27$) (Brooks and Esque, 2002), but the increase was only significant in the low and middle elevation zones below 1,280 m (4,200 ft) ($r^2 = 0.32$, 1980 to 2001) (M. Brooks, unpublished data). A few areas burned three separate times during this 15-year interval. The increase in fire frequency was due to increased number of fires caused by humans, since the number of lightning-caused fires remained constant (Brooks and Esque, 2002). Another major contributor to increased fire frequency was a general increase in fine fuel loads caused by heightened dominance of non-native annual grasses beginning in the late 1970s (e.g. Hunter 1991) and continuing on through the 1990s (M. Brooks personal observation), probably the result of above-average rainfall from 1976 to 1998 (Hereford et al. in press). Although most fires were small and started along roadsides, most of the large fires occurred in remote areas far from major roads, and were typically started by lightning (Brooks and Esque, 2002).

Major Ecological Zones

In this section we describe the basic fire ecology of the predominant plant species in each ecological zone. We also discuss patterns of postfire succession, and interactions between plant communities, fire behavior, and fire regimes. More details on the fire ecology of a wider range of desert species can be found in other recent publications (Brown and Smith 2000, Esque and Schwalbe 2002).

Low Elevation Desert Shrubland Zone

This zone includes two primary vegetation types. Alkali sink vegetation occurs on poorly drained saline and/or alkaline playas, flats, and fans approximately -80 to 1,200 m (-63 to 3,937 ft) throughout all the ecological sections. Plant communities include iodine bush-alkali scrub, allscale-alkali scrub, Mojave saltbush-allscale scrub, and saltgrass meadow (Rowlands 1980). Creosote bush scrub vegetation occurs 0 to 1,200 m (0 to 3,937 ft) on well-drained flats, fans, and upland slopes of the Mojave, Colorado, and Sonoran Desert ecological sections. However, only the lower elevations below about 900 m (2,953 ft), where perennial plant cover is relatively low, are typical of the low elevation desert shrubland zone. Plant communities include creosote bush scrub, cheesebush scrub, succulent scrub (Rowlands 1980).

Fire responses of important species

Most shrubs in the low elevation zone do not survive after being completely consumed by fire (Humphrey 1974, Wright and Bailey 1982) (Table 16.3), but since many fires in this zone are patchy and of low intensity, plants frequently survive in unburned islands. Low fire temperatures in interspaces, and high temperatures beneath woody shrubs, likely results in relatively higher seedbank mortality for annual plants that frequent beneath-shrub than interspace microhabitats (Brooks 2002). A few perennial species that evolved to resprout after natural flooding disturbances often resprout after burning, such as desert willow (*Chilopsis linearis*), catclaw (*Acacia greggi*), smoke tree (*Dalea spinosa*), and cheesebush (*Hymenoclea salsola*) (Table 16.3). Cheesebush can have almost 100% survival rates even after being totally consumed by fire (Table 16.4). Cacti are usually only scorched during fires, as flames propagate through their spines but the stems do not ignite due to their high moisture content. Individuals with high levels of scorching typically die from uncontrolled desiccation that occurs postfire. Cactus regeneration can occur from resprouting of partially scorched plants, or rooting of fallen unburned stem fragments, but less frequently from establishment of new seedlings.

Table 16.3. Fire responses of some dominant plant species in the southeastern deserts bioregion.

Lifeform	Predominant Type of Fire Response ¹		Species
	Sprouting	Individual	
Conifer	none	killed	bristlecone pine, limber pine, pinyon pine, Utah juniper
Hardwood	fire stimulated	top-killed	shrub live oak, salt cedar*, honey mesquite, willows
	fire stimulated	underburned	Fremont cottonwood,
Shrub	fire stimulated	top-killed	catclaw acacia, smoke tree, desert willow, fourwing saltbush, cheesebush, rubber rabbitbrush, spiny hopsage, antelope bitterbrush
	none	killed	Shadscale, blackbrush, creosote bush, brittlebrush, white bursage, snakeweed, cliffrose
Stem and Leaf	none	killed	Cacti
Succulents	fire stimulated	top-killed	Mojave yucca, banana yucca, Joshua tree ²
Herb	fire stimulated	top-killed	bulbs
	none	killed	annual forbs
Grass	fire stimulated	top-killed	perennial grasses (e.g. Galleta grass, Indian ricegrass, desert needlegrass, fountain grass*)
	none	killed	annual grasses (e.g. red brome*, Mediterranean grass*, cheatgrass*, six-week fescue).

¹ varies depending on fire intensity and percentage of plants consumed

² after initially resprouting, Joshua trees often die within 5 years if most or all of their foliage was scorched or consumed

* non-native species

Table 16.4. Survival rates of perennial shrubs after being unburned (0% of living biomass burned), scorched (1-10% of living biomass burned) or consumed (11-100% of living biomass burned) during 2.25ha (5.6acre) fires in August 1995 at three low elevation shrubland sites (M. Brooks, unpublished data).

species ^a	sample size	% survival ^b			notes on fire behavior ^c
		year 1	year 4	year 8	
Central Mojave Site					
white bursage					Fire did not spread from ignition points. Therefore, the litter beneath each shrub, but not the shrub itself, was ignited. Most creosote bushes were consumed, because accumulated dead branches beneath them provided supplemental surface fuels that increased flame residency time beneath them, and ladder fuels that helped carry fire up into the creosote bush canopies.
unburned	n=20	100	100	85	
consumed	n=20	20	20	10	
creosote bush					
unburned	n=25	100	100	100	
scorched	n=4	25	25	25	
consumed	n=21	0	0	0	
Southern Mojave Site					
creosote bush					Fire spread rapidly from a few ignition points and burned 50% of the site. Few shrubs were consumed due to low fuel loads beneath creosote bushes, and low cover of finely textured sub-shrubs.
unburned	n=25	100	100	100	
scorched	n=13	77	70	62	
consumed	n=12	8	8	8	
Western Mojave Site					
white bursage					Fire spread slowly from multiple ignition points, and burned 50% of the site. Most shrubs were consumed due to high fuel loads beneath creosote bushes and the presence of many finely textured sub-shrubs.
unburned	n=10	100	100	90	
consumed	n=10	20	20	20	
cheesebush					
unburned	n=10	100	100	60	
consumed	n=10	100	100	80	
creosote bush					
unburned	n=25	100	100	100	
scorched	n=8	88	75	75	
consumed	n=17	12	12	12	
Anderson wolfberry					
unburned	n=20	100	100	90	
scorched	n=5	100	100	80	
consumed	n=20	75	75	50	

^a Dominant perennial plant species at each site. Not all were represented by both scorched and consumed plants. white bursage = *Ambrosia dumosa*, creosote bush = *Larrea tridentata*, cheesebush = *Hymenoclea salsola*, Anderson wolfberry = *Lycium andersonii*.

^b Shrub survival was defined as possessing live leaf tissue, either on unburned or resprouted stems, when sampled during May of 1996, 1999, and 2003.

^c Additional descriptions of the fires and study sites are reported in Brooks 1999.

The most frequently encountered and dominant shrub in this zone, creosote bush (*Larrea tridentata*), can have 25 to 80% survival rates 8 years postfire when it is only scorched (1 to 10% biomass loss), and 0 to 12% survival rates by year 8 when it is consumed by fire (11 to 100% biomass loss) (Table 16.4). Individuals with slight to moderate scorching displayed 30 to 40% survival in the Sonoran Desert in Arizona (Dalton 1962), and in general, fire intensity and duration is inversely correlated with sprout reproduction (White 1968).

The wide range in survival rates among creosote bushes appears to be associated with their variable physiognomy and variable fuel loads beneath their canopies and across the landscape, which translate into variable fire intensity and vertical continuity from surface to canopy fuels. Individuals with canopies in the shape of inverted cones tend to occur in water-limited environments (De Soyza et al. 1997), resulting in relatively low fuel loads beneath their canopies and across the landscape and a relatively low probability of being completely consumed by fire. In contrast, individuals with hemispherical canopies that extend to the ground tend to occur in less water-limited environments (De Soyza et al. 1997), resulting in higher fuel loads beneath their canopies and across the landscape and a higher probability of being completely consumed by fire. Resprouting in creosote bushes also probably varies throughout the extensive range of this species, especially at ecotones with vegetation types that support more frequent burning. For example, moderate (O'Leary and Minnich 1981, Brown 1984) to high (Brown 1984) rates of postfire resprouting were reported at the ecotone of the western Colorado Desert with shrubland vegetation in the Peninsular ranges.

Fire regime-plant community interactions

This is the zone that Humphrey (1974) was primarily referring to when he stated that in desert shrublands "...fires are a rarity, and the few fires that do occur cause little apparent damage to the various aspects of the ecosystem..." (pp. 337). This is largely because fuels are discontinuous and characterized by a sparse 8 to 15% cover of woody shrubs, and the large interspaces between shrubs are mostly devoid of vegetation, inhibiting fire spread (Fig. 16.1). A recent summary of fire regimes of the United States (Schmidt et al. 2002) assumed that Küchler's "barren vegetation type" (Küchler 1964), which covers most of the low elevation desert shrubland zone, is mostly devoid of vegetation and therefore fireproof. However, 9% of fires and 7% of the total area burned between 1980 and 2001 occurred within the barren vegetation type in the California desert bioregion. Thus, fires do occur in the low elevation desert shrubland zone, although not as frequently and over less area than in the other zones of the desert bioregion.

Fire behavior and fire regimes in this zone are affected primarily by the ephemeral production of fine fuels from annual plants. Years of high winter and spring rainfall can increase continuity of fine fuels by stimulating the growth of annual plants that fill interspaces and allow fire to spread (Brown and Minnich 1986, Schmid and Rogers 1988, Rogers and Vint 1987, Brooks 1999). Native annuals that produce some of the most persistent fuelbeds include the annual grasses six-weeks fescue (*Vulpia octoflora*) and

small fesue (*Vulpia microstachys*), and the large forbs fiddleneck (*Amsinckia tessellata*), tansy mustard (*Descurania pinnata*), and lacy phacelia (*Phacelia tanacetifolia*), compared to a whole suite of smaller native forbs (119 species, Brooks 1999). Infrequently, successive years of high rainfall may have allowed these native annuals to build up fine fuel loads sufficient to carry fire across the interspaces between larger perennial plants. Low elevation fires carried by high loads of native annuals typically only burn dead annual plants and finely-textured sub-shrubs, leaving many of the larger woody shrubs such as creosote bush unburned. Thus, the historic fire regime was likely characterized by relatively small, patchy, low intensity surface fires, and a truncated long fire return interval (Table 16.5).

Table 16.5. Fire regime classification for desert shrubland zones.

Desert shrubland zone	<i>Vegetation Type</i>			
		Low elevation shrubland	Middle elevation shrubland and grassland	High elevation shrubland and woodland
	<i>Temporal</i>			
	Seasonality	Spring-summer-fall	Spring-summer- fall	Summer-early fall
	Fire Return Interval	Truncated long	Long	Long
	<i>Spatial</i>			
	Size	Small	Moderate to large	Moderate to Large
	Complexity	High	Multiple	Low to Moderate
	<i>Magnitude</i>			
	Intensity	Low	Moderate	Moderate to High
	Severity	Moderate	Moderate to high	High
	Fire Type	Surface	Passive crown to active crown	Active crown

The invasion of non-native annual grasses into the desert bioregion introduced new fuel conditions. Species such as red brome and Mediterranean grass (*Schismus arabicus* and *S. barbatus*) provide more persistent and less patchy fine fuelbeds than do native annual plants, breaking down more slowly and persisting longer into the summer and subsequent years (Brooks 1999). These new fuel conditions have the potential to increase

the size, decrease the complexity, and shorten the time interval between desert fires, although fire intensity will likely decrease because fine herbaceous non-native fuels are replacing coarse woody native fuels. These fire regime changes have occurred over a small fraction of the low elevation ecological zone, and fire regimes over the vast majority of this zone still are within the historical range of variation.

Mediterranean grass is the most widespread and abundant non-native annual grass in the low elevation shrubland zone, although red brome may predominate under large shrubs or in the less arid parts of this ecological zone. Mediterranean grass has fueled fires as large as 41 ha (100 acres) (Bureau of Land Management DI-1202 records), and interspace fuel loads of as little as 112 kg/ha (100 lbs/acre) are sufficient to carry fire (Brooks 1999). Because these fires burn with low intensity, soil heating is negligible and most woody shrubs are left unburned.

The recent spread of Sahara mustard (*Brassica tournefortii*) throughout the low elevation shrublands has caused concern that this invasive mustard may introduce a significant new fuel type to the desert bioregion. During years of high rainfall this invasive annual can exceed 1 m (3.3 ft) in height with a rosette of basal leaves 1 m (3.3 ft) across, and even moderately sized plants can produce as many as 16,000 seeds (M. Brooks unpublished data). Plants can remain rooted and upright through the summer fire season, and when they finally do break off they blow like a tumbleweed and lodge in shrubs or fencerows, accumulating piles of fuels similar to Russian thistle (*Salsola* spp.). There are no records of fires specifically caused by Sahara mustard in the desert bioregion, however, the combination of this species with red brome in the understory helped fuel a 20.2 ha (50 acre) fire in creosote bush scrub in northwest Arizona (M. Brooks, personal observation). During the 5 years after this fire, Sahara mustard and red brome have come to dominate this site while the native creosote bush has yet to show signs of recovery.

Non-native annual plants that evolved in other desert regions will likely be most successful at persisting in the California desert bioregion. For example, Mediterranean grass and Sahara mustard respectively evolved in the arid Middle East and Northern Africa, and they have also successfully established in the desert bioregion (Brooks 2000, Minnich and Sanders 2000). At three sites in the western Colorado Desert, these non-native species successfully persisted through two major droughts, which occurred during the end of the 1980s and 1990s (R. Minnich, unpublished data). Their cover values in 1983, 1988, and 1990 through 2001 were comparable or higher than those of the non-native forb red-stemmed filaree (*Erodium cicutarium*), which is a poorer fuel source for fires (Brooks 1999), and compared to all native forbs combined.

Middle Elevation Desert Shrubland and Grassland Zone

This zone includes five primary vegetation types. The upper elevations of creosote bush scrub that generally occur at 900 to 1,200 m (2,953 to 3,937 ft) and contain higher perennial plant cover than the lower elevations of this vegetation type. Joshua tree woodland occurs on well-drained loamy, sandy, or fine gravelly soils of mesas and gentle slopes from 760 to 1,300 m (2,493 to 4,265 ft) in the Mojave Desert and Southeastern Great Basin sections. Shadscale scrub occurs on heavy, rocky, often calcareous soils with

underlying hardpan from 1,000 to 1,800m (3,281 to 5,906 ft) in the Mono, Southeastern Great Basin, and Mojave Desert sections. Blackbrush occurs on well drained, sandy to gravelly often calcareous soils from 1,000 to 2,000m (3,281 to 6,562 ft) in the southern Mono, Southeastern Great Basin and Mojave Desert sections. Desert scrub-steppe vegetation types are intermixed with a wide range of other plant communities from the low to the high elevation ecological zones, but they are most common in the middle elevation zone. Indian ricegrass scrub-steppe and desert needlegrass scrub-steppe typically occur where winter rainfall predominates within creosote bush scrub (Rowlands 1980). Big galleta scrub-steppe typically occurs in creosote bush scrub below 1,000 m (3,281 ft), and in Joshua tree woodland and blackbrush scrub above 1,000 m (3,281 ft).

Fire responses of important species

Higher fuel loads and more continuous fuelbeds in the middle elevation ecological zone result in higher intensity fires and higher frequency of top-killing in plants than in the low elevation zone. However, more species in this zone are likely to resprout after being top-killed. Perennial grasses such as desert needlegrass (*Achnatherum speciosum*), galleta grass (*Pleuraphis rigida* and *P. jamesii*), and Indian ricegrass (*Achnatherum hymenoides*) readily resprout after burning (Table 16.3). Spiny menodora (*Menodora spinescens*) and joint-fir (*Ephedra* spp.) often survive fire because their foliage does not readily burn. In contrast, some shrub species such as blackbrush (*Coloegyne ramosissima*) and winterfat (*Kraschennikovia lanata*) rarely survive burning.

Blackbrush is one of the more flammable native shrubs in the desert bioregion, due to its high proportion of fine fuels and optimal packing ratio. In the rare case that only a portion of a shrub is consumed, it may survive and resprout from the root crown. This resprouting was observed within the first few postfire years (Bates 1984), and these resprouts were still evident 20 years later (M. Brooks personal observation), at a site in the Mono section near Bishop, California. It is commonly thought that blackbrush stands take centuries to recover (Bowns 1973, Webb et al. 1988). However, analyses of historical photographs from Joshua Tree National Park and southern Nevada indicate that blackbrush stands can recover within 50 to 75 years (Minnich 2003, M. Brooks unpublished data), although other historical photographs from other locations do not indicate recovery within this time interval (M. Brooks, unpublished data). It seems probable that the ability of blackbrush to resprout after burning varies across its wide geographic range which extends from the Colorado Plateau and southern Great Basin on through the Mojave Desert.

Yucca species such as Joshua tree (*Yucca brevifolia*), Mojave yucca (*Yucca schidigera*), banana yucca (*Yucca baccata*), and Our Lord's candle (*Yucca whipplei*) are typically scorched as flames propagate through the shag of dead leaves that line their trunks. The relatively small size and more optimal packing ratio of dead Joshua tree leaves compared to dead Mojave or banana yucca leaves, increases the frequency at which they are completely burned. This may partly explain why Joshua trees are more frequently killed by fire. All four yucca species readily resprout after fire, but Joshua tree resprouts are often eaten by herbivores or otherwise die soon after burning. Postfire recruitment of new Joshua trees is infrequent, and likely occurs during years of high rainfall. No seedlings or saplings were observed in burns <10 years old, and only <10 individuals/hectare were

present on burns >40 years old in Joshua Tree National Park (Minnich 2003). Joshua tree populations along the extreme western edge of the desert bioregion often resprout and survive more readily after fire than those further east (M. Brooks personal observation). A cycle of relatively frequent fire and resprouting can result in short, dense clusters of Joshua tree clones, such as those found near Walker Pass, in the western end of the Antelope Valley, and in pinyon-juniper woodlands at ecotones with the Transverse ranges. High resprouting rates of Joshua trees in these areas may have evolved in local ecotypes that became adapted to relatively high fire frequencies at the ecotone between the desert bioregion and more mesic ecosystems to the west.

Fire regime-plant community interactions

Some of the most continuous native upland fuels in the desert bioregion occur at the upper elevations of this zone, especially in areas dominated by blackbrush (Fig. 16.2). Invasive annual grasses have contributed to increased fire frequencies since the 1970's (Brooks and Esque, 2002), although the native perennial vegetation in this zone can at times be sufficient alone to carry fire during extreme fire weather conditions (Humphrey 1974). Between 1980 and 2001, 49% of all fires and 45% of total area burned occurred in Küchler's desert shrubland vegetation type, which is roughly analogous to the middle elevation ecological zone.

At the lower elevations within this zone, where creosotebush is co-dominant with a wide range of other shrubs and perennial grasses, fire spread is largely dependent on high production of fine fuels filling interspaces during years of high rainfall (Brown and Minnich 1986, Schmid and Rogers 1988, Rogers and Vint 1987, Brooks 1999). At higher elevations within this zone, where blackbrush is often the primary dominant plant, fire spread is not so dependent on the infilling of shrub interspaces during years of high rainfall and fire occurrence does not vary as much inter-annually compared to lower elevations (M. Brooks unpublished data). Thus, the historic fire regime was likely characterized by relatively moderate to large sized, patchy to complete, moderate intensity, surface to crown fires, and a long fire return interval (Table 16.5).

The post-fire response of plant communities in blackbrush scrub is illustrative of the general responses of other desert scrub communities in the middle and high elevation ecological zones. Blackbrush fires remove cover of woody shrubs which is soon replaced by equivalent cover of herbaceous perennials and annual plants (Brooks and Matchett 2003). Alien species such as red brome, cheatgrass, and red-stemmed filaree typically increase in cover after fire, but only if rainfall is sufficient to support their growth and reproduction. Recovery of blackbrush stands may occur within 50 years (Minnich 2003, M. Brooks, unpublished data), but perhaps more typically take over 100 years (Webb et. 1988, Bowns 1973).

Red brome is the dominant invasive grass at middle elevations in the California desert bioregion. This invasive grass produces higher fuel loads and fuel depths than does Mediterranean grass, and accordingly produces longer flame lengths that carry fire into the crowns of large woody shrubs more readily, producing more intense fires (Brooks 1999). Cover of red brome can become greater and more continuous after fire, promoting recurrent fire (Sidebar 16.1, this volume). This positive invasive plant/fire regime cycle

(*sensu* Brooks et al. 2004) has shifted fire regimes outside of their historical range of variation in some regional hotspots (Brooks and Esque 2002), although fire regimes in most of the middle elevation zone are probably similar to historical conditions.

The recent invasion of the non-native annual grass African needlegrass (*Stipa capensis*) into the ecotone between the Colorado Desert and the Peninsular ranges in the 1990s has helped fuel at least one 243 ha (600 acre) fire (R. Minnich, personal observation). There are early indications that this species can survive relatively dry years, suggesting that it may spread and become another source of fine fuels that may further alter fire regimes in the desert bioregion.

High Elevation Desert Shrubland and Woodland Zone

This zone includes three primary vegetation types. Sagebrush scrub occurs in 1,100 to 2,800 m (3,600 to 9,186), although it can extend to 3,800 m (12,467 ft) in the White Mountains. Pinyon-juniper woodland occurs 1,300 to 2,400 m (4,265 to 7,874 ft), and can reach 2,700 m (8,858 ft) in the White Mountains. Both vegetation types occur in the Mono, Southeastern Great Basin, and Mojave sections. Among the pinyon-juniper vegetation types, the Utah juniper – single-leaf pinyon association is the most widespread, occurring in the Mono, Southeastern Great Basin, and eastern Mojave Desert ecological sections of California (Minnich and Everett 2001). The California juniper – single-leaf pinyon association occurs along the desert slopes of the Transverse ranges at the edge of the Mojave Desert section, with California juniper dominating below 1,700 m and single-leaf pinyon dominating above. Desert chaparral is the least prevalent of the major vegetation types in this ecological zone. It occurs on the middle slopes of the Transverse ranges adjacent to the Mojave Desert, and the Peninsular ranges adjacent to the Colorado Desert, below the mixed conifer forests, and in the same general elevation range as sagebrush scrub and pinyon-juniper woodland.

Fire responses of important species

Relatively high fuel loads result in high fire intensity, but plant mortality rates can vary widely among species. Wyoming big sagebrush (*Artemisia tridentate* ssp. *wyominensis*) is typically killed by fire, but it often re-establishes readily from wind-dispersed seeds. Cliffrose (*Purshia mexicana*) is typically killed by fire, whereas its close relative, antelope bitterbrush (*Purshia tridentata*), exhibits highly variable responses to fire, sometimes resprouting (Table 16.3). Interior chaparral species, such as *Quercus cornelius-mulleri*, *Q. turbinella*, *Cercocarpus betuloides*, *Arctostaphylos glauca*, and *A. glandulosa*, *Nolina* spp., either resprout or reseed soon after fire, but lower rainfall and sparser vegetation cover results in less frequent fire and slower recovery rates than is typical of cis-montane chaparral.

Pinyon pine (*Pinus monophylla*, *P. edulis*) and juniper (*Juniperus osteosperma*, *J. californica*) are typically killed by fire, but these woodlands can re-establish after 100+ years of fire exclusion. Juniper typically re-establishes from seed sooner than pinyon pine. Initial establishment of single-leaf pinyon pine appears to be delayed 20 to 30 years by sun

scald and/or freeze/thaw soil heaving until the establishment of the shrub layer and young juniper trees which act as nurse plants (Wangler and Minnich 1996). The first pinyon recruits establish within the canopies of nurse plants, often near root axes. The establishment of a pinyon pine canopy after about 75 years eventually reduces freeze-thaw processes, setting off a chain-reaction of spatially random recruitment throughout old burns. Pinyons develop complete canopy closure after 100 to 150 yr which is accompanied by a decline in the surface vegetation, due apparently to shrub senescence and shade stress.

Fire regime-plant community interactions

Fuel continuity is similar to that of the middle elevation zone, but the fuels are generally more woody and difficult to ignite. In addition to high plant cover, the prevalence of steep slopes in this ecological zone facilitates the spread of fire. Due to the high biomass of woody fuels created by juniper and pinyon pine, and to a lesser extent sagebrush (*Artemisia* spp.), bitterbrush, cliffrose, and scrub oak (*Quercus turbinella*), the fires that do start are among the most intense encountered in the desert bioregion. Between 1980 and 2001, 33% of fires and 45% of the total area burned occurred in Küchler's sagebrush, juniper-pinyon, and chaparral vegetation types which are characteristic of the high elevation ecological zone.

Fire spread can occur most any year in sagebrush steppe, although it is more likely when fine fuel loads (especially cheatgrass and red brome) are high following years of high rainfall, or during periods of high winds and low relative humidity. Fires are patchy to complete, moderate intensity passive crown to crown fires, depending the continuity of the woody shrub fuels. Fire spread in pinyon-juniper woodlands is most probable when live fuel moisture and relative humidity are low and winds are high. When fires did historically occur, they were mostly large, intense crown fires, burning through woodland crown fuels. At the interface between sagebrush steppe and pinyon-juniper woodland, a surface to passive crown fire regime is the norm, as fire spreads through woody and herbaceous surface fuels and occasionally torches woodland fuels, especially younger trees. The historic fire regime was likely characterized by relatively large, patchy to complete, moderate intensity surface to crown fires, and a long fire return interval (Table 16.5).

Sagebrush stands generally require 30 to 100 years to recover following fire (Whisenant 1990). Where cheatgrass has dramatically shortened fire return-intervals, especially in the lower elevation Wyoming big sagebrush communities, sagebrush steppe has been converted to non-native annual grassland (Sidebar 16.1, this volume). In the higher elevation mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) communities, this type conversion is much less common, since the native shrubs and perennial grasses recover much more rapidly after fire.

Fire suppression coupled with removal of fine fuels by livestock grazing has allowed pinyon-juniper woodlands to encroach on sagebrush steppe across much of the western United States (Miller and Tausch 2001), including the Mono section of the desert bioregion. However, it is less likely that woodland encroachment has occurred in the more arid hot desert regions, due to low primary productivity rates. Recent resampling of 1929 to 1934 California Vegetation Type Map (VTM) survey plots reveal no significant changes in woodland densities at the western edge of the hot desert regions (Wangler and Minnich

1996). Pinyon-juniper woodlands adjacent to the Transverse ranges have experienced long periods between stand-replacement fires both before and after fire suppression began (fire rotation periods, ~450 years; Wanger and Minnich 1996).

Fires in pinyon-juniper woodlands are least frequent in open stands at lower elevations and more frequent in dense forests at higher elevations, in response to changing productivity and fuel accumulation gradients with increasing elevation and rainfall. The upper elevation ecotones between pinyon-juniper woodlands and mixed conifer forest are typically very narrow, due to truncated disturbance gradients related to fire behavior and stem mortality (Minnich 1988). The thin bark of pinyon pine prevents their survival in the frequent surface fire regime typical of mixed conifer forests. Alternatively, postfire surface fuels appear to lack sufficient biomass to support short-period burns, and as canopy closure occurs in pinyon and juniper woodlands, surface fuel loads and continuity are further reduced. Thus, a historical discontinuity in fire return intervals probably existed along the ecotones between mixed conifer forests and pinyon woodlands in which understory surface fires at high elevations shift to long-period stand-replacement crown fires at lower elevations in response to differences in stand structure, fire behavior, and tree survivorship (Minnich 1988).

Desert Montane Woodland and Forest Zone

There are two primary vegetation types in this ecological zone. Bristlecone-limber pine forests occur on well-drained, shallow, dolomitic soils from 2,600 to 3,800 m (8,530 to 12,467 ft) in the Inyo, White, Panamint, Funeral, and Grapevine mountains. Alpine fell-fields occur above timberline, primarily in the White Mountains. Small white fir forest enclaves also occur on north-facing slopes from 1,900 to 2,400m (6,234 to 7,874 ft) in the New York, Clark, and Kingston mountains of the Mojave Desert section (Rowlands 1980).

Fire responses of important species

The flagship tree species of this ecological zone, bristlecone pine and limber pine (*Pinus flexilis*), have thin bark which makes them susceptible to mortality during fires (Table 16.3). Although most individuals are struck by lightning by the time they are 1,000 years old, strikes may not result in the entire tree burning, since many old individuals have scars resulting from multiple lightning strikes. The presence of ancient bristlecone pine individuals is testimony to the historical infrequency of fire. As a result, most plant species in this zone are not adapted to recovery from fire, although species associated with other periodic natural disturbances such as from colluvial erosion may be able to resprout after burning.

Fire regime-plant community interactions

Fuels are very discontinuous, but in contrast to the low elevation zone, ephemeral production by annuals during years of high rainfall adds very little to the fuel bed, due to shallow soils, low temperatures, and a short growing season. As a result, surface fires are extremely rare, and most fires that do occur spread through the crowns of pines only

during extreme fire weather conditions, but even these fires are very small <1ha (2.5 acres). Between 1980 and 2001, <1% of all fires and total area burned occurred in Küchler's great basin pine, alpine meadows-barren, and mixed conifer vegetation types characteristic of the desert montane ecological zone.

Low productivity results in very low fuel loads and continuity in the desert montane forests. Except on steep, north-facing canyons, heavy fuels are widely spaced and fine fuels are low and relatively unflammable, making it difficult to carry fire in this landscape. Thus, the historic fire regime is characterized by truncated small, patchy, variable intensity, passive crown fires, and a truncated long fire return interval (Table 16.6).

Table 16.6. Fire regime classification for the desert montane woodland, and riparian woodland/oasis zones.

Desert montane woodland, and riparian woodland/oasis zone	<i>Vegetation Type</i>		
		Desert montane woodland	Riparian woodland/oasis zone
	<i>Temporal</i>		
	Seasonality	Summer-early fall	Spring-summer-fall
	Fire Return Interval	Truncated long	Short to moderate
	<i>Spatial</i>		
	Size	Truncated small	Small to moderate
	Complexity	Moderate	Low
	<i>Magnitude</i>		
	Intensity	Multiple	High
	Severity	Multiple	Multiple
	Fire Type	Passive Crown	Passive to active crown

Desert Riparian Woodland and Oasis Zone

Riparian woodlands occur primarily along the Colorado and Mojave river corridors adjacent to low elevation shrublands in the southern desert region. Other examples can be found in the Amargosa Gorge, Whitewater River, Andreas Canyon, and Palm Canyon. In the northern desert region, riparian woodlands occur along the Owens and Walker rivers and the many creeks along the east slope of the Sierra Nevada Mountains. Oasis woodlands occur in isolated stands such as the Palm Canyon, Thousand Palms, and Twentynine-palms oases in the Colorado Desert section.

Fire responses of important species

Woodland dominants such as Fremont cottonwood (*Populus fremontii*) honey mesquite (*Prosopis glandulosa*), and willows (*Salix* spp.) typically resprout after being topkilled (Table 16.3). However, resprouting individuals and seedlings are susceptible to mortality during recurrent fires. Oasis species such as Washington fan palm (*Washingtonia filifera*) benefit from frequent, low-intensity fire, which reduces competition for water from other plants growing at the surface, and allow new seedlings to become established.

Fire regime-plant community interactions

Fuel characteristics and fire behavior are extremely variable, due to the wide range of vegetation types that characterize the riparian zone. In general fuels are typically continuous and fuel loads high, but fuel moisture content is also often high. Fires may not carry except under extreme fire weather conditions. Thus, the historic fire regime is characterized by small to moderate sized, complete, high intensity passive to active crown fires, and a short to moderate fire return interval (Table 16.6).

In riparian woodlands the invasives saltcedar (*Tamarix* spp), and less frequently giant reed (*Arundo donax*), create ladder fuels that allow fire to spread from surface fuels of willow (*Salix* spp.), saltbush (*Atriplex* spp.), sedge (*Carex* spp.), reed (*Juncus* spp.), and arrow weed (*Pluchea sericea*) into the crowns of overstory Fremont cottonwood trees, top-killing them. After an initial fire, these invasive quickly recover and surpass their pre-fire dominance, promoting increasingly more frequent and intense fires which, can eventually displace most native plants (Sidebar 16.2, this volume).

In palm oases, Washington fan palms depend on surface fire to clear understory species and facilitate recruitment. However, these sites can be pre-empted by saltcedar as it rapidly recovers after fire. The ladder fuels saltcedar creates can also carry fire into the crown of Washington fan palms, increasing the incidence of crown fires (Sidebar 16.2, this volume)

Management Issues

Fuels Management

The deserts of southwestern North America are one of the fastest growing regions in terms of human populations in the United States. As human populations increase, so to do the number of people living at the wildland-urban interface, which complicates fire management in many ways (Chapter 19, this volume). Increasing human populations can also potentially change fuel characteristics, through increased air pollution which can increase deposition rates of atmospheric nitrogen, and potentially increase fine fuel loads (Brooks 2003). Burgeoning human populations can also increase the introduction rates of new plant species that could add new fuel components and fire hazards to the region (Chapter 22, this volume). Since fire spread is mostly limited by the availability of contiguous fuels, fuel management can be a very important tool for fire managers in the California desert bioregion, even though the areas in which it is used may be a small percentage of the total region.

Herbaceous fuel management

The fuel component of greatest concern in the desert bioregion is the continuous cover the non-native annual grasses red brome, cheatgrass, and Mediterranean grass that appear during years of high rainfall. Although populations of these non-native annual plants and their resultant fine fuel loadings wax and wane with annual and multi-decadal fluctuations in rainfall (Sidebar 16.1, this volume), they have changed fire behavior and fire regimes in many parts of the desert bioregion, especially in the low elevation ecological zone where their presence is almost a prerequisite for large fires.

Despite all the concern surrounding the non-native species already dominating the desert bioregion, new grass invaders such as fountain grass (*Pennisetum setaceum*), buffelgrass (*Cenchrus ciliaris*), and African needlegrass, and invasive mustards such as Sahara mustard, may pose additional fire hazards in the future. For example, in the Sonoran Desert, buffelgrass invasion coupled with frequent fire has converted desert scrub to non-native grassland in Mexico (Búrquez et al. 2002), created fuels sufficient to carry fire in Arizona, and recently appeared in southeastern California (M. Brooks personal observation). Land managers who once lamented the damage caused by fires fueled by red brome in southern Arizona are even more concerned now about the potential effects of buffelgrass (S. Rutman, Organ Pipe Cactus National Monument, personal communication). Buffelgrass is currently being considered for addition to the Arizona Department of Agriculture, Noxious Weed List, due primarily to its ability to alter fire regimes (E. Northam, personal communication). Thus, fine fuels management should be closely tied to invasive plant management, because the predominant plant invaders in the southern part of the desert bioregion are relatively flammable herbaceous species (Brooks and Esque, 2002). This is important both from the perspective of managing invasive plant fuels that are currently present, and preventing the establishment of new invasive plants that may

change fuel structure and potentially cause even greater fire management problems in the future.

Livestock grazing has been mentioned as a possible tool for managing fine fuels in the desert bioregion (Brooks et al. 2003, Minnich 2003). It may temporarily reduce fine fuel loads, and be effective for managing fuels in specific areas such as within the wildland urban interface. However, grazing may also reduce cover of late seral native plants and replace them with non-native annual and other early seral plant species (Brooks et al. 2003) that can be more flammable. Grazing treatments must be applied with attention to the potential responses of all dominant plant species, both in the short term based on the phenologic stage during which they are grazed, and in the long term based on their life history characteristics and inter-relationships among species.

Woody fuel management

Where native plant cover is sufficient to carry fire without the addition of fine fuels from non-native plants, coarse woody fuels are the major concern of fire managers. In the central and southern parts of the desert bioregion, blackbrush intermixed with perennial grasses, Joshua trees, and juniper produce the right mix of high fuel continuity, fuel loads, and fuel packing ratio that can cause large intense fires with frequent spotting ahead of the flaming front. Although infrequent, intense, stand-replacing fires are a natural part of blackbrush shrubland ecology, these types of fires are not desirable when they occur near human habitations, or where they may damage cultural resources such as historical buildings or pre-historical sites. Once these fires start, they often require indirect firefighting tactics to suppress, which complicates efforts to protect specific areas from burning. As a result, land managers and scientists are testing ways to reduce the chances of extreme fire behavior in this vegetation type where it occurs between Joshua Tree National Park and the communities of Yucca Valley and Joshua Tree (M. Brooks et al., unpublished data). They are comparing the effects of fire and mechanical blackbrush thinning on subsequent fuel conditions, fire behavior, and plant community structure. The goal is to find tools that will allow managers to manipulate fuel characteristics to reduce fire hazards near areas identified for protection from fire, while having minimal negative ecological effects, such as increased dominance of invasive non-native plants.

Sagebrush and pinyon-juniper fuels are the primary focus of fuel management in the northern parts of the desert bioregion, especially in the Mono section. Sagebrush intermixed with perennial grasses is generally considered to be a greater fire hazard than the blackbrush communities described above. A century or more of fire exclusion, livestock grazing, and climate change can also result in encroachment by pinyon-juniper woodlands into sagebrush steppe (Miller and Tausch 2001). This has been documented in the northeast bioregion of California (Schaefer et al. 2003), and has also occurred where rainfall is relatively high in the desert bioregion at the ecotone of the Great Basin desert with the Sierra Nevada Mountains (Anne Halford, botanist, BLM-Bishop Field Office). Dense stands of mature trees in that area increase the chance of intense, stand-replacing, crown fire. Unfortunately, these same mature woodlands are desirable for use as homesites, especially in the Mono section, complicating the implementation of fuels management treatments and the protection of homes during fires. Millions of hectares are

planned for fuels reduction in the western United States (<http://www.fireplan.gov>), and much will involve thinning of smaller size classes of pinyon and juniper trees to allow surface fuels to increase, and moderate intensity surface fires to return to the ecotone between pinyon-juniper woodlands and sagebrush steppe. Because very little is known about the effectiveness of these treatments in changing fire behavior or the potential ecological effects of these treatments, a research project was recently begun to quantify the effects of pinyon and juniper thinning on subsequent fuel condition, fire behavior, and ecosystem variables at a site in northwestern Arizona (M. Brooks et al., unpublished data).

Where sagebrush and pinyon-juniper vegetation interface in the southern desert sections, they are either at high elevations far from major roads and human habitations, or they contain surface fuels of insufficient amount and continuity to carry fire. These stands only burn under extreme fire weather conditions. Analyses of aerial photographs and VTM survey data from the 1930s show no evidence of pinyon-juniper expansion in the southern parts of the California desert region (R. Minnich, unpublished data). Accordingly, management of pinyon-juniper fuels is not advisable in this region, except where needed for specific cultural resource or safety reasons.

Fire Suppression

There is specific concern about the effect of fire suppression activities on the federally threatened desert tortoise where it occurs in low and middle elevation zones (Sidebar 16.3, this volume). More generally, fire suppression in desert wilderness areas became a significant issue after the California Desert Protection Act (1994) applied this designation to many new areas. Wilderness areas often encompass mountain ranges in the desert bioregion, where locally high fuel loads from both native and non-native plants, and steep slopes, facilitate the spread of fire. Fire suppression options are generally more limited in these areas by the constraints outlined in wilderness management plans, and often the primary tactic is to wait for fire to spread down slope and attempt to stop it along pre-existing roads. This can result in large portions of desert mountain ranges burning during a single event. The question is, which causes greater ecological damage, activities associated with aggressive fire fighting (e.g. construction of hand or bulldozer control lines, fire retardant drops) or large-scale, sometimes recurrent, fire occurring where fires were historically small and infrequent? We recommend that suppression be a high priority where fire frequency has been recently high in regional hotspots and non-native grass fire cycles have become locally established (Brooks and Esque 2002), where local populations of non-native plants may be poised to expand their range and landscape dominance following fire (mostly in the middle elevation ecological zone), or where there are other management reasons to exclude fire. Otherwise, a let burn policy for natural fires may be appropriate.

Postfire Restoration

Burn Area Emergency Rehabilitation (BAER) teams have developed postfire restoration/rehabilitation plans after the large fires that have recently occurred in the Mono section (e.g. Cannon and Slinkard fires), and further south in the desert bioregion at its ecotone with the Transverse and Peninsular ranges (e.g. the Juniper Complex and Willow fires). Much of this effort is focused on protecting watersheds from soil erosion, and one of the common tools is the seeding of rapidly growing plants (Sidebar 20.1, this volume). In general, seeding treatments establish more readily in the cold deserts than in the hot deserts, although relative establishment rates and the ecological effects of seeding in these two regions have not been experimentally compared.

Postfire seeding may also be used to compete with and reduce the cover of invasive grasses associated with the grass-fire cycle (Sidebar 16.1, this volume). The idea is to replace highly flammable species such as cheatgrass with less flammable seeded species. Non-native perennial grasses such as crested wheatgrass (*Agropyron desertorum*) have been used to compete with and reduce cover of cheatgrass in Great Basin sagebrush steppe. However, there has been a recent move toward using native species in postfire seeding, which may not have the same effect as non-native perennial grasses in suppressing the growth of non-native annual grasses such as cheatgrass. A current study is evaluating the relative effectiveness of non-native versus native perennial grasses to compete with and reduce cover of cheatgrass after fires in sagebrush steppe in the Mono section, and at sites in the Great Basin and Colorado Plateau (M. Brooks, unpublished data).

Fire Management Planning

One of the biggest challenges in fire management planning is determining desired future conditions to use as management goals. In cases where historical fire regimes can be reconstructed (e.g. ponderosa pine forest), the natural range and variation of historical fire regime characteristics may be a realistic and appropriate target. However, management goals may be elusive where historical fire regimes cannot be easily reconstructed, such as in the desert bioregion where one must rely on indirect inferences.

Fire histories alone may not be enough to establish management goals when protection of specific natural or cultural resources are the primary management goal, or where plant invasions have changed the rules of the game. For example, if plant invasions have shifted fuel characteristics outside of their natural range of historical variation, then restoration of historical fire regimes may be impossible without first dealing with the invasive plants that are at the root of the problem (Brooks et al. 2004). Although it appears that fire regimes, and at least woody fuel conditions, across much of the desert bioregion may be within their historical range of variation, it is difficult to quantify the impact that non-native plant invasions have had, aside from recognizing that fire regimes have been altered dramatically in some regional hotspots (Brooks and Esque 2002). Further complicating this process are the effects of potential future changes in rainfall patterns (Hereford et al. in press), and levels of atmospheric CO² (Mayeaux et al. 1994) and

nitrogen deposition (Brooks 2003), on fuel conditions and fire regimes. All of these potential variables need to be considered when determining fire management goals in the desert bioregion.

The recent mandate by federal land management agencies to create fire management plans for all management units has resulted in a flurry of activity as new plans are drafted and old plans are revised. In many cases, plans developed for desert management units are supported by relatively few scientific studies, due to the paucity of fire research that has been conducted in the desert regions of North America. Decisions on when and where fuels should be managed, fires should be suppressed or allowed to burn, or post-fire restoration projects should be implemented, are difficult to make given the limited data available. Recent reviews have attempted to provide land managers and others with current information on desert fire ecology and management (Brooks and Pyke 2001, Brooks and Esque 2002, Esque and Schwalbe 2002, Esque et al 2002, Brooks et al. 2003). Along these same lines, a primary purpose of this desert bioregion chapter is to provide additional information that can be used in the development of fire management plans in the deserts of southwestern North America.

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SIDEBAR 16.1. Ecological Effects of Non-native Annual Grasses

by Matthew Brooks and Richard Minnich

Non-native annual grasses in the genera *Bromus*, *Avena*, and others have become dominant components of many grasslands, shrublands, woodlands, and forests in western North America during the 20th century. These invasions have negatively affected native plant species by directly competing with them for limiting soil nutrients and water, and by altering ecosystem properties such as fuel characteristics and fire regimes. The positive feedback between non-native grass dominance and increased fire frequency, or the “grass/fire cycle” (D’Antonio and Vitousek 1992), is the most clearly understood and well-documented example of the more general “invasive plant/fire regime cycle” (Brooks et al. 2004).

Fire frequencies that have increased beyond their historical range and variation can have dramatic and far-reaching ecological effects. For example, invasion of the non-native cheatgrass (*Bromus tectorum*) has altered fuelbed characteristics and shortened fire return intervals from 30 to 100 years to 5 years in areas of the Great Basin (Whisenant 1990). This new fire regime promotes the dominance of cheatgrass over native species, resulting in large-scale conversions of high diversity, native sagebrush steppe to low diversity, non-native annual grassland. This vegetation change has negatively affected animals that require sagebrush steppe for forage and cover such as the sage grouse (*Centrocercus urophasianus*) (Sidebar 11.2, this volume), and prey species such as black-tailed jackrabbits (*Lepus californicus*) and the Paiute ground squirrel (*Spermophilus mollis*) which are important for golden eagles (*Aquila chrysaetos*) and prairie falcons (*Falco mexicanus*) (Knick and Rotenberry 1995, Knick et al 2003). Although similar large-scale higher-order effects have not been documented in the Mojave, Colorado, or Sonoran deserts, non-native grass/fire cycles have degraded habitat for the desert tortoise (*Gopherus agassizii*) in localized hotspots within these desert regions (Brooks and Esque 2002, Sidebar 16.3 this volume).

Invasive plant/fire regime cycles represent ecosystem shifts to alternative stable states which will likely persist unless fuels, climate, or ignition patterns significantly change (Brooks et al. 2004). For example, non-native annual grasses such as cheatgrass and medusahead (*Taeniatherum caput-medusea*) persist in cold desert regions like the Great Basin because rainfall is typically sufficient to support reproduction during any given year. Although the fuelbeds they create may only significantly affect fire behavior following years of high rainfall, their populations will likely persist even during years of low rainfall. As a result, non-native grasses and the altered fire regimes they cause are now relatively permanent features in many parts of the cold desert region.

In contrast, the hot desert regions of the Mojave, Colorado, and Sonoran deserts receive less annual rainfall than the cold desert regions, increasing the chances of population crashes of non-native annual grasses such as cheatgrass and red brome (*Bromus rubens*). Rainfall events as small as 5 mm (2 in) can stimulate their germination, and when there is little subsequent rainfall, the plants often die before reproducing (M. Brooks and R. Minnich, personal observations), potentially depleting the soil seedbank. This is probably why red brome became locally extinct at two low elevation desert sites after the late 1980s drought, and at one low elevation site after the late 1990s drought (R Minnich, unpublished

data). However, extirpation of red brome did not occur at many higher elevation desert sites following these same drought periods (M. Brooks, personal observation). Broad-scale responses by non-native grasses to droughts indicate that they are typically not regionally extirpated and can recover to ecologically significant numbers relatively quickly in hot desert regions. For example, after the end of the approximately 30-year mid-century drought (Hereford et al. in press) red brome density and biomass jumped 700% and 150% respectively between the last year of the drought (1975) and the first year of higher rainfall (1976), and by 1988 the increase above 1975 levels reached 15,646% for density and 1,596% for biomass at a Mojave Desert/Great Basin ecotone in southern Nevada (Hunter 1991). During this time interval, density and biomass of native annuals decreased (Hunter 1991), while the frequency and size of fires across the Mojave Desert steadily increased (Brooks and Esque 2002,). In addition, the shorter 1987-1991 drought was followed in 1993 by one of the biggest fire years in the 1980-2001 agency fire record for the hot desert regions, and the spread of many of these fires was facilitated by substantial fine fuelbeds of red brome and cheatgrass.

Thus, non-native annual grasses will not likely ever become extirpated from the hot desert regions under the current climate regime, although their landscape dominance and effects on fire frequency and behavior will undoubtedly continue to be highly episodic in response to rainfall. Non-native grass/fire cycles have already become established in some localized hotspots within the hot desert region (Brooks and Esque 2002). The extent of area affected by these vegetation and fire regime type-conversions may expand during periods of high rainfall in the future, although most of this expansion will probably be confined to the middle elevation desert shrubland and grassland ecological zone. Below the middle elevation zone, extreme drought conditions will cause more frequent population crashes of red brome and cheatgrass and thus limit their influence on fire regimes, and above the middle elevation zone native woody plants and perennial grasses are the primary factors affecting fire regimes.

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SIDEBAR 16.2. Saltcedar Invasions Can Change Riparian Fire Regimes

by Tom Dudley and Matthew Brooks

Saltcedar (*Tamarix ramossissima*) was brought to North America in the early 1800's by European colonists as a horticultural plant, and by the early 1900's it became widely used to provide windbreaks and erosion control along railways and other erosion-prone sites. Its ability to tolerate periodic drought and harsh soil conditions helped insure its establishment persistence where other species failed. It was recognized as an invader of desert watercourses around the 1920's, and with the advent of water control and diversion projects took advantage of the altered conditions to expand its range during the middle and latter part of the century (Robinson 1965).

Saltcedar is deciduous and produces a fine-structured, water-repellent litter layer that is highly flammable in late summer and fall. Because stand densities can be very high, and litter is slow to decompose, a nearly-continuous layer of surface fuels can develop which carries fire throughout the stand (Busch and Smith 1992). The standing trees are also flammable, and can carry fire from surface fuels up into the canopies of native riparian trees. These fuel characteristics can create a frequent, high intensity, crown fire regime where an infrequent, low to moderate intensity, surface fire regime previously existed. After burning, saltcedar stump-sprouts readily and benefits from nutrients released by fire, whereas native riparian plants such as cottonwood and willow do not resprout as vigorously (Ellis 2001). Recurrent high intensity fire may lead to monoculture stands of saltcedar. Thus, saltcedar has turned many watercourses from barriers of fire movement to pathways for fire spread.

As stands of saltcedar increase in density and cover, native cottonwood and willow trees decrease. In some cases this is coincident with changing environmental conditions that do not favor the native species (e.g. decreased water tables caused by water diversion projects; Everitt 1998), but in other cases it is clear that saltcedar is responsible for the

decline in native trees, directly through competition and indirectly through altered fire regimes (Busch and Smith 1995). Because it provides lower quantity and quality of shade, forage, and insect prey species, wildlife generally avoid large stands of saltcedar in preference for native stands (Shafroth et al. in press). This includes numerous threatened and declining riparian birds which find better nesting and feeding resources on native trees. In addition, saltcedar can have higher evapotranspiration rates than native trees, potentially reducing water tables (Sala et al. 1996). All of these symptoms of saltcedar invasion have caused major management problems in southwestern riparian ecosystems.

Mechanical and chemical methods are typically used to manage saltcedar, however they can be very expensive (\$300 - \$6,000/ha.; Shafroth et al. in press), their effectiveness is often limited and temporary, and they can have other undesirable ecosystem effects. After more than a decade of pre-release testing, a leaf-feeding beetle from Eurasia, *Diorhabda elongata*, has been experimentally released in several western states as a biological control agent against saltcedar (Dudley et al. 2000). At one site in northern Nevada this beetle defoliated approx. 2 ha in 2002, and spread to defoliate over 400 ha in 2003. The physiological stress experienced by defoliated plants may lead to lowered live fuel moisture, and definitely increases the amount of dead wood and foliage. In the short term, this biocontrol may increase the chance of high-intensity fire, but in the long run the conversion of saltcedar stands back to native riparian woodlands will likely reduce fire hazards.

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SIDEBAR 16.3. Fire Effects on the Desert Tortoise (*Gopherus agassizii*)

by Matthew Brooks and Todd Esque

Changing fire regimes threaten 12 of the 40 major tortoise species worldwide (Swingland and Klemens 1989). Only general habitat destruction is listed as a threat for more species (23 of 40 species). In general, tortoises are poorly adapted to fire because they evolved in arid or semi-arid habitats where fire was historically rare. The desert tortoise (*Gopherus agassizii*), Mojave population, is a Federally Threatened species listed partly because of threats posed by fire.

Fires can kill desert tortoises, especially fires that occur in the spring and early summer when they are most active above-ground throughout their range (Esque et al. 2002). Years of high rainfall produce the profuse annual plant growth that is required for desert tortoise reproduction, but it also contributes to fire occurrence, especially at the low and middle elevation zones within the desert tortoise range. Thus, years when growth and reproduction are expected to be greatest can be coincident with increased fire occurrence. Although, mortality from individual fires is generally considered insignificant for wildlife populations compared to the habitat changes that can follow, loss of a few individuals may be catastrophic for local populations of species that are already in decline (Esque et al. 2003).

Fires can also affect desert tortoises indirectly, by changing habitat structure and plant species composition. Loss of cover sites that provide protection from the sun and predators, and loss of native forage plants are specific examples of the potential negative effects of fire (Brooks and Esque 2002, Esque et al. 2002). Individual fires may have relatively small indirect effects within desert tortoise habitat, since they are often patchy, leaving unburned islands of native vegetation. In contrast, recurrent fires pose a much greater threat, as they often burn through previously unburned islands of vegetation, and can produce broad landscapes devoid of shrub cover and dominated by non-native annual grasses. These conditions are currently focused within a number of regional hotspots in the desert bioregion (Brooks and Esque 2002).

When fighting fires that occur within desert tortoises habitat in the low and middle elevation zones, land managers follow guidelines developed to reduce the chance of killing desert tortoises such as not burning out unburned habitat islands when feasible, checking under tires before moving vehicles, and walking ahead of vehicles when they are required to travel off-road (Duck et al. 1998). Results of firefighting activities in desert tortoise habitats have proven that the benefit of fighting fires in desert tortoise habitat far outweighs the potential danger of damage to habitats and tortoise populations when appropriate guidelines are followed (Duck et al. 1998).

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